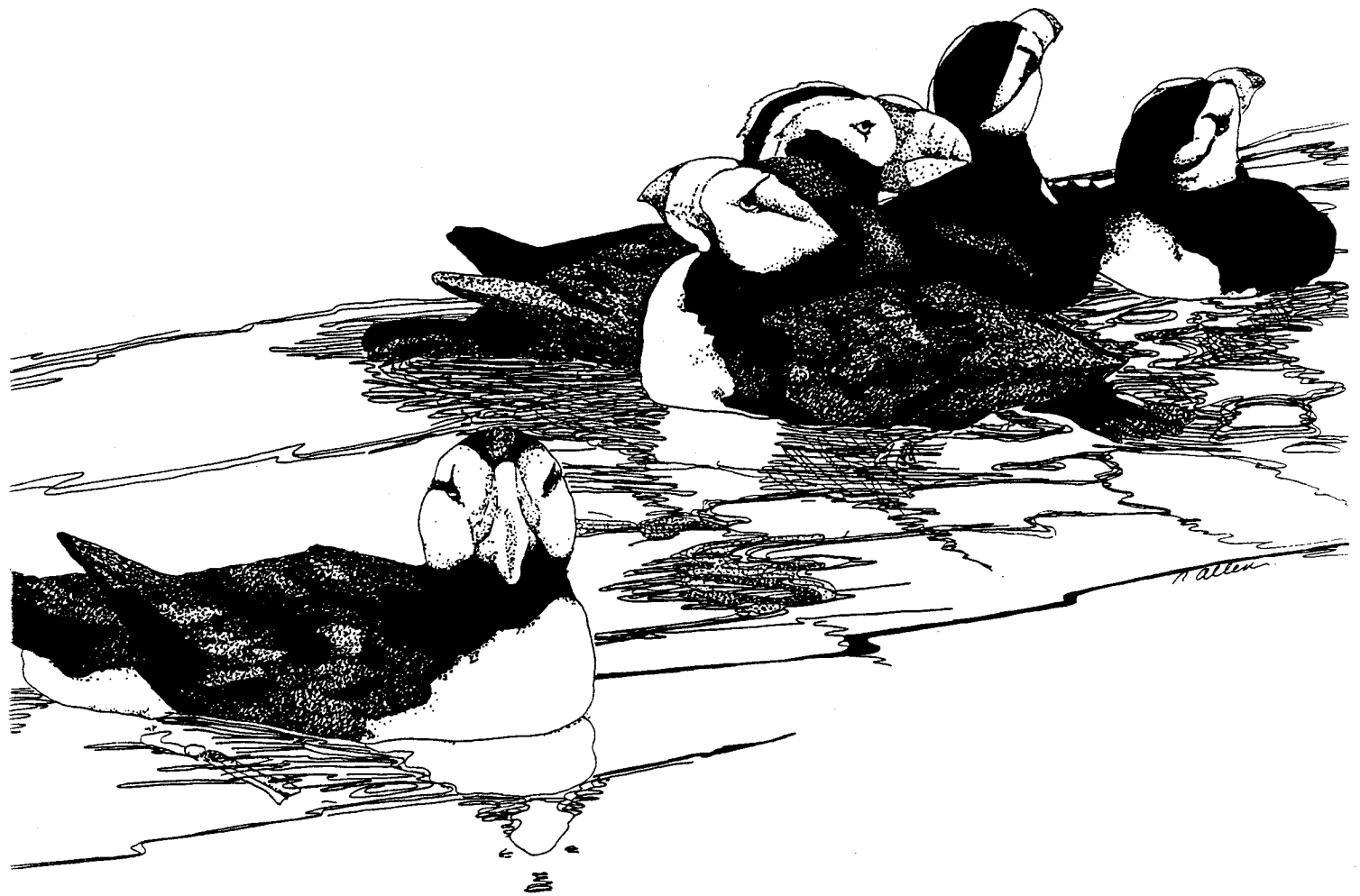


The Effects of Oil on Wildlife:

Research, Rehabilitation, and General Concerns



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Effects of Oil Pollution on Marine Bird Populations

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ABSTRACT

Worldwide oil pollution has killed millions of marine birds in this century but it has been difficult to directly link these losses to population declines. Estimated bird losses from acute spills and chronic pollution are not precise because we usually do not know the proportion of birds killed at sea that are detected on beach surveys or the origin of those birds. Data required to assess effects on populations (abundance, distribution, productivity, recruitment and mortality rates) are inadequate or absent for many species. Local populations may sometimes be devastated by oil pollution, but whether these losses are biologically significant to global populations, especially in light of natural or human-induced sources of mortality, is debatable. In this paper, we review the evidence for effects of oil on marine bird populations, discuss four case histories, and address the debate concerning short- and long-term effects on avian populations.

INTRODUCTION

Marine birds are the most conspicuous victims of oil spills at sea and the only organisms that are killed regularly by oil pollution on a scale likely to affect local, regional or global populations. In the northern hemisphere, diving birds

like loons, grebes, pelecaniforms, seaducks, and alcids are most affected because they spend much of their lives in contact with the sea surface and because their populations are concentrated in waters with chronic oil pollution.

For these vulnerable species, biologists and resource managers need to establish the degree to which oil mortality affects populations. If populations are seriously at risk, then the threat of oil pollution needs to be mitigated in the future. If populations are not threatened by oil, effort should be focused on other threats to seabirds: e.g., mortality from gill-nets, habitat loss, hunting, etc. To address this question we: (1) review evidence for known and suspected effects of oil pollution; (2) examine for case studies; and (3) evaluate what constitutes a "significant biological effect" and whether such effects are likely to be important, particularly with respect to marine bird populations in North America.

PROBLEMS IN DEMONSTRATING CAUSE AND EFFECT RELATIONSHIPS

Unequivocal demonstration of a direct cause and effect relationship between oil pollution and fluctuations in size of marine bird populations has been difficult because:

(1) Estimates of total mortality from either acute or chronic oil pollution events often are imprecise and few birds that are killed at sea by oil are recovered (Ford et al. 1987). We may have confidence in the minimum estimates derived from carcasses recovered on land, but reliable extrapolations from carcass counts to total mortality are rarely possible. Knowledge of the long-term effects of sublethal oiling is meagre.

(2) Marine birds are difficult to census and few breeding colonies and their constituent species are being monitored carefully enough to detect losses following an oil spill (Nettleship and Evans 1985). Certain species, like Common Murres (*Uria aalge*), are relatively easy to monitor because they breed on observable cliff-ledges in a limited number of large colonies (Nettleship 1976, Birkhead and Nettleship 1980, Harris et al. 1983). But other species are extremely difficult to census because they are either non-colonial breeders with widely dispersed nests (e.g., Harlequin Ducks *Histrionicus histrionicus*, or loons), or they nest in burrows or crevices in the ground (Nettleship 1976) and in some cases may visit their breeding site only at night (e.g. Cassin's Auklet *Ptychoramphus aleuticus*). For many of these problem species, only crude estimates of population size and status exist. At-sea monitoring of bird populations using shipboard and aerial survey techniques has been adequate for describing gross distributional patterns for many species. But such pelagic surveys are of limited value for assessing either population size or status owing to large temporal and spatial variability in numbers and distribution (Brown et al. 1975). Rare exceptions include some species that concentrate outside the breeding season at the same locations year after year (e.g., seaducks, loons, and grebes; Campbell et al. 1978, Camphuysen 1989) or occur close to shore in well-defined marine habitats (e.g., Marbled Murrelet *Brachyramphus marmoratus*; Sealy and Carter 1984).

(3) Even where populations are carefully monitored, it is often difficult to ascribe changes in abundance to specific causes except in obvious circumstances, such as a spill close to a colony (Stowe 1982a,b). We need much more information about how environmental variability, natural mortality, recruitment, and density-dependent effects influence marine bird population dynamics. Oil mortality also can not be considered independently from other sources of anthropogenic and natural mortality (Dunnet 1982).

(4) Most mortality from oil spills occurs in winter when marine birds have dispersed from their breeding locations and are mixed with birds from other regions (Brown et al. 1973, Andrews and Standing 1979, Hope-Jones et al. 1982). Thus, oil mortality at any single location often involves several populations, which makes it difficult to detect impacts on single populations or colonies. Where species are resident throughout the year, effects of oil on populations may be more conspicuous.

EVIDENCE FOR POPULATION EFFECTS

Direct observation: beach surveys and banding studies

Recoveries of bird carcasses on coastal beaches indicate that tens of thousands of marine birds have often been killed in single oil spills (Anker-Nilssen and Rostad 1981, National Research Council [NRC] 1985, Anker-Nilssen et al. 1988, Piatt and Lensink 1989), and chronic pollution has killed millions of birds world-wide during this century (Tuck 1961, Tanis and Morzer Bruyns 1968, Croxall 1975, 1977; Bourne 1976, Nettleship 1977, Stowe 1982c, Croxall et al. 1984, Stowe and Underwood 1984, Evans and Nettleship 1985, Piatt et al. 1985, Hooper et al. 1987, Camphuysen 1989). Bird mortality from oil was probably extreme during the war years of 1939-1945 when oil pollution was rampant. Oil mortality was also more common prior to the 1960's, after which improved technologies and new regulations for the shipment and transfer of oil reduced chronic pollution (Bourne 1968, 1976; Ainley and Lewis 1974, Croxall 1975).

Given the difficulties of measuring marine bird population trends and the incompleteness of records on the frequency and magnitude of oil pollution at sea, it has been difficult to directly link this long term oil pollution to declining marine bird populations. However, in many regions where chronic oil pollution has occurred — e.g., the English Channel, North Sea, Skagerrak, Baltic Sea, Gulf of St. Lawrence, and coastal United States (Atlantic and Pacific) — many populations of marine birds have either disappeared or diminished drastically in size during this century (Evans 1984, Nettleship and Evans 1985). Other sources of human disturbance (e.g., hunting, eggging, toxic chemicals, commercial fisheries) have sometimes contributed heavily to early population declines, but, in the presence of chronic oil mortality, many populations have never recovered despite the elimination or significant reduction of those other mortality factors (Nettleship and Evans 1985).

More direct evidence for population effects has been collected during the last 50 years from banding studies and beach surveys. It is clear that bird mortality from oil pollution was, and still is, widespread and persistent throughout the north Atlantic and Pacific oceans. The problem is most severe in cold, northern waters (Fig. 1) near oil tanker traffic lanes where oil persists longer and vulnerable species are most abundant (Bourne 1976, NRC 1985). Of the tens of thousands of dead birds washed up on beaches every year — particularly alcids, sea ducks, loons, and other divers — a high proportion are oiled. Long-term mean oiling rates of 50-90% are not unusual in western Europe and eastern Canada, although means of less than 20% have been recorded elsewhere in North America (Fig. 1, Kuyken 1978, Andrews and Standring 1979, Stowe 1982c, Piatt et al. 1985, Simons 1985, Stenzel et al. 1988, Camphuysen 1989). Often, oiling is the most frequent cause of death when cause of death can be determined (Stenzel et al. 1988). Moreover, analyses of band recoveries in Britain indicate that: (1) on average, 18-28% of all alcid mortality since the 1930's may be attributed to oil pollution (Lloyd 1974, Mead 1974, Harris 1984, Hudson 1985); (2) actual proportions killed by oil were very much higher in certain regions (e.g., North Sea and south coast); and (3) numbers killed by oil are underestimated by a factor of two or more in band recovery analyses (Baillie and Stowe 1984). In summary, the data clearly indicate that in many areas with chronic oil pollution, mortality from oil has long been a principal cause of death for several marine bird species.

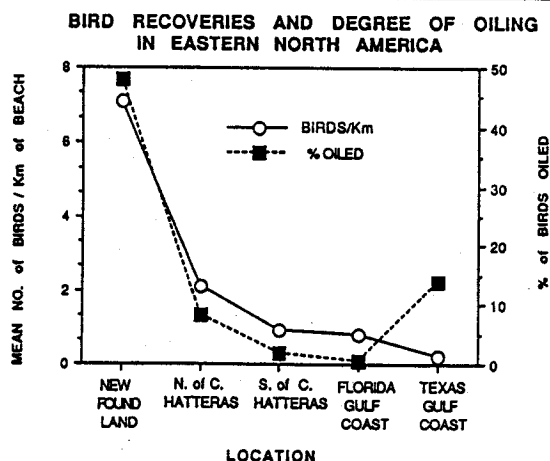


Figure 1. Numbers and degree of oiling of birds recovered on beach surveys in eastern North America (from Piatt et al. 1985). Data from U.S. provided by Malcolm Simons (Atlantic and Gulf Coast Beached Bird Survey Project). Averages based on 5 yrs for north of Cape Hatteras (1975-1979), 4 yrs for south of Cape Hatteras and Florida (1976-1979), 3 yrs for Texas (1977-1979), and 4 yrs for Newfoundland (1980-1982, 1984). Birds/km is the mean number of birds found per km of beaches checked on each monthly survey, averaged for the entire year.

From carcass drift experiments, it is known that the number of birds recovered on beach surveys usually represents only a small fraction of the total actually killed at sea (Table 1). Most birds sink during the days and weeks following mortality at sea, and the proportion that make it to shore depends largely on wind speed and direction following the spill, proximity to land, and prevailing currents (Bibby and Lloyd 1977). For reasons probably related to buoyancy, wind profile, size, and color, gull carcasses are recovered on beaches at much higher rates than alcid carcasses (Table 1). Search effort is also an important factor, and recovery rates of carcasses on beaches are higher following well-publicized oil spills (e.g., Hope-Jones et al. 1970) than following chronic small spills. Once on shore, carcasses often disappear rapidly as they are buried in sand or debris, or removed by scavengers (Ford et al. 1987, Page et al. 1990, Piatt et al. 1990). Oil mortality far offshore or away from populated regions is usually not detected or quantified (Brown 1973), though mortality is known to be sometimes high (Brown and Chardine, this volume; Canadian Department of Fisheries and Oceans, unpublished records). This factor alone may lead to serious underestimates of oil mortality, because a significant proportion of most seabird populations aggregate far at sea on banks and shelf-edges frequented by ships.

Thus, counts of dead carcasses on beaches only provide minimum numbers killed by oil from which to extrapolate an estimate of total mortality. Various models have been constructed that estimate total mortality by integrating the movement and area of oil at sea with known bird distributions — however, these models need to be calibrated using beached bird data before they can be trusted to reflect true mortality levels. Carcass recovery rates, the most critical data required for extrapolation, are extremely variable (Table 1), and depend largely on the specific spill scenario, local environmental conditions following a spill, and the species involved.

There has been a tendency by certain researchers (Bourne 1976, Dunnet 1982, Clark 1984) to reject high estimates of oil mortality because of "tenuous" extrapolations from beached carcass recoveries. However, conservative (or "defensible") estimates are equally tenuous and more commonly reported (e.g., Brown et al. 1973, Hope-Jones et al. 1978, Page et al. 1990, Piatt et al. 1990). We agree with Croxall (1975) that estimates of bird mortality in the hundreds of thousands or millions for the North Atlantic provided by Tanis and Morzer Bruyns (1968) and others (Ampleford and Brown 1959, Tuck 1961, Nelson-Smith 1972, Anker-Nilssen and Rostad 1981, Piatt et al. 1985,

Camphuysen 1989) are not unreasonable. An estimate that at least a million marine birds, mostly alcids, are killed annually by oil pollution in the north Atlantic and Pacific oceans is not unrealistic. If anything, this value might be considered low based on existing information. Overall, there is a need for more drift experiments and more detailed modelling exercises (simulations that estimate oil mortality from sinking rates, losses on beaches, and trajectories of carcasses, e.g., Ford et al. 1987, Page et al. 1990). These data would greatly improve our ability to assess the true magnitude of bird mortality, especially from large, acute oil spills.

Demographic effects and population dynamics

Banding studies and examinations of oiled carcasses indicate that virtually all age classes and sexes of marine birds are affected by oil. Individual spills may affect one predominant age- or sex-class if populations are segregated on breeding or wintering areas (e.g., Brown et al. 1973, Camphuysen 1983). Over the long term, however, it appears that for most of the vulnerable taxa (loons, seaducks, alcids) about 40-60% of the birds killed by oil have been adults (Lloyd 1974, Mead 1974, Harris 1984, Simons 1985, Camphuysen 1989).

The population biology of some species affected by oil has been relatively well-studied. For example, the life history of Common Murres has been documented extensively in the north Atlantic and Pacific (e.g., Tuck 1961, Nettleship and Birkhead 1985, Ainley and Boekelheide 1990, Hatch and Hatch 1990). Like many other seabirds affected by oil (Dunnet 1982), murres have a low reproductive rate (one-egg clutch), reach sexual maturity slowly (first breeding: 4-6 years of age), and display a low adult mortality (usually <10% per annum). These life-history parameters make it difficult for most seabird populations to recover quickly from losses. Some other species, like eider ducks, have a higher reproductive potential (multi-egg clutches), begin breeding earlier, and as a consequence may be able to compensate more quickly for losses (Dunnet 1982, Reed 1986).

It is possible to mathematically model murre population dynamics using measured population parameters (Leslie 1966, Ford et al. 1982, Samuels and Lanfear 1982, Hudson 1985). This indirect approach is valuable because it allows predictions to be made of how populations should respond to a variety of perturbations that would otherwise be impossible to measure in field situations. Models indicate that the time required for murre populations to recover from oil spill losses

(or losses from any cause) depends to some extent on the age-classes affected. It takes longer for recovery when adult rather than immature birds are killed. Recovery times are lengthy when all age classes are killed in proportion to their local abundance. For example, a population loss of 50% spread among all age groups can require on the order of about 70 years for recovery, and perhaps twice that time if population growth is adjusted for density dependence at higher population levels (Samuel and Lanfear 1982). One-time losses during the breeding season affect not only individual breeding birds, but also their current and future offspring and mates so that one-time loss effects carry over in subsequent years (Ford et al. 1982, Takekawa et al. 1990). Chronic low-level pollution may produce permanent changes in survivorship and fecundity schedules by increasing adult mortality rates and reducing average longevity (Ford et al. 1982). Small decreases in fecundity or adult survival produce large increases in recovery time. Murre breeding success varies with the density of breeding birds on cliff ledges (Birkhead 1985) and there may be minimum "threshold" densities below which colony extinction is inevitable in the absence of immigration (Hudson 1985).

Oil mortality versus natural mortality

Much of the uncertainty about the importance of oil mortality to bird populations arises from one central argument. It is believed that under usual conditions, marine bird populations are limited by resources and a 5-20% annual adult mortality rate occurs naturally. Such mortality is balanced in stable populations by continuous production and recruitment of sub-adults into the adult breeding population. To some degree, recruitment and mortality are compensatory and the equilibrium between them is constantly shifted by a host of ecological and anthropogenic factors (Birkhead and Furness 1985). Short- and long-term fluctuations in abundance, sometimes including one-time catastrophic losses, are normal in healthy populations, although the magnitude and frequency of fluctuations varies between species. Tens of thousands of seabirds occasionally die *en masse* from starvation or disease (Tuck 1961, Bailey and Davenport 1972, Piatt et al. 1990) or by drowning in gill-nets (Piatt and Nettleship 1987). If mortality from natural causes (starvation, disease, predation), and anthropogenic causes (gill-nets, hunting, and oil) are interchangeable as means of population reduction, then oil mortality, by itself, may not be biologically significant in a stable population even if it accounts for a major proportion of total mortality. But if oil

mortality acts in addition to natural mortality, then we must consider that significant levels of oil mortality will lead to population decline.

Dunnet (1982) addressed this issue for seabird populations in Britain. He compared conservative estimates of birds killed annually by oil in the 1970's (ca. tens of thousands) with expected annual losses of immature (ca. 463,000) and adult (ca. 194,000) seabirds from natural mortality (based on known population sizes and life tables). He concluded that oil losses of this magnitude were not large compared to natural losses. He further argued that because some seabird populations in oil-polluted areas of Great Britain were recently increasing, losses from oil pollution were not biologically significant to those populations. This view has been echoed elsewhere (Clark 1984, NRC 1985, Baker et al. 1990) with little, if any, real justification.

Several inconsistencies in this argument, however, lead us to a very different conclusion. For instance, if the number of birds dying from natural causes far exceeded the number killed by oil, then why were so few uniled dead birds found on beaches? The fact that only a few thousand uniled birds were found on British beaches when 100,000's were presumably dying offshore substantiates experimental evidence that a very small fraction of birds killed at sea ever make it to shore. Thus, estimates of oil mortality based on beached carcass counts are so minimal that they may not even indicate the correct order of magnitude of the kill toll. In contrast, calculations of expected levels of natural mortality are probably reasonable long-term averages. Furthermore, Dunnet (1982) did not incorporate into his argument the fact that many British birds killed by oil at sea wash up on continental European beaches (Bourne 1976) and numbers retrieved there actually exceed numbers retrieved on British beaches (Camphuysen 1989).

Another important omission in the Dunnett (1982) review is the fact that about 40-60% of birds killed by oil were breeding adults whereas the bulk of birds dying annually from natural causes were juveniles and immatures. Dunnet (1982) did not consider age-specific effects in his argument, but clearly losses of adults are far more important than losses of immature birds. Relative to this point is also the fact that most oil mortality takes place in mid to late winter after much of the natural culling of immature and "unfit" individuals has already taken place. Oil does not discriminate between "fit" and "unfit" members of populations. Owing to this, it seems unlikely that mortality from oil can be substituted directly for natural mortality.

Perhaps the most serious point that

Dunnet (1982) failed to mention was that while some populations of British seabirds did increase in the 1960' and 1970's, many of those increases reflected partial recovery by relict populations under optimal natural conditions from substantial losses incurred throughout this and previous centuries (NRC 1985, Nettleship and Evans 1985, Harris and Wanless 1988). Presumably, the potential rates of recovery of these colonies would have been much greater in the absence of oil mortality. It is noteworthy that many colonies in regions subjected to heavy chronic pollution in the early part of this century and more recently have never recovered (Brun 1979, Stowe 1982b, Evans 1984, Nettleship and Evans 1985).

In summary, our collective experience indicates that oil mortality is possibly additive to, and cannot be considered independently from, other sources of mortality. If overall natural mortality is density-dependent and adequately compensated for with recruitment, then stable populations at carrying capacity may be able to absorb "considerable" losses from oil mortality and still remain stable or even increase for some period of time. However, this is not equivalent to concluding that oil mortality has no biological effect on populations. Clearly, oil mortality will be more important where it affects small, stressed, or reduced populations (Dunnet 1982, Camphuysen 1989); as the following case histories illustrate.

CASE HISTORIES

The following studies provide examples of potential and known effects of oil pollution on marine bird populations. Much of this data is presented and discussed in detail elsewhere, so only the main points are summarized here.

The North Sea: Bempton Cliffs

In April of 1977, an oil spill near a Common Murre colony at Bempton Cliffs, on the east coast of England, affected over 1,400 resident murrees and probably some others from colonies to the north (Stowe 1982a). Census plots at the Bempton Cliffs colony had been monitored almost every year from 1972 to 1981. Counts of birds on plots in 1977 after the spill were about 20% lower than those recorded in 1976, differences that were statistically significant ($P < 0.05-0.001$). However, numbers of murrees increased by about 40% in 1978 ($P < 0.05-0.001$) reaching a level that exceeded numbers recorded before the 1977 spill. Similarly, whole-colony counts showed that murre numbers had dropped from 12,200 individuals in 1975 to 9,224 in 1977 (-24%) and increased to 13,250 in

1978 (+44% between 1977-78; +9% against the baseline). The spill appeared to have little long-term effect on the population as it steadily increased by about 7.5% per annum over the period of study (1972-1981). Stowe (1982a) concluded that the decline in numbers in 1977 may have resulted largely from the temporary absence of potential breeders disturbed by the spill (i.e., temporary desertion, loss of mates, toxin-induced or behavioral inhibition of breeding). The "recovery" in 1978 probably reflected the return of mature birds that deferred breeding in 1977 supplemented by an influx of new recruits and immigrants from other colonies (Stowe 1982a).

This example typifies the situation for many British seabird colonies in the North Sea, where several seabird populations increased since intensive monitoring began in the 1970's (Stowe 1982b) despite chronic oil pollution (Stowe and Underwood 1984). However, similar increases were not observed concurrently at most colonies on the south and west coasts of Britain (Stowe 1982b, Evans 1984). Furthermore, populations in the North Sea have either stabilized or started to decline again since the mid-1980's, possibly in response to changing food abundance and environmental conditions (Harris and Wanless 1988, 1990; Harris 1991). At the same time, there has also been a marked increase in the frequency of oiled and unoled birds found on beaches in the southern North Sea since the mid-1980's (Camphuysen 1989). In context of these changing conditions, oil mortality may now (again) be a significant mortality factor for marine birds in the North Sea (Camphuysen 1989). Similarly, populations of seabirds off Norway and in the Barents Sea may once have been able to withstand considerable oil mortality (Barrett 1979), but a recent crash in forage fish stocks and changing environmental conditions (Vader, et al. 1990) make these populations more vulnerable to oil pollution.

The English Channel: Les Sept Isles

Only a few hundred Atlantic Puffins (*Fratercula arctica*) now breed in France along the Brittany coast on the English Channel (Nettleship and Evans 1985). The population was formerly much larger (Fig. 2). At Les Sept Isles, there were 10,000-100,000 breeding pairs in the 1860's-1870's (Henry and Monnat 1981), reduced later to about 10,000-15,000 birds in 1900 owing to hunting and eggging, and to only 1,000 birds by 1911 (Yeatman 1976). In 1912, Les Sept Isles was provided legal protection and numbers increased to 3,000 pairs in 1921 and to about 7,000 pairs by 1927. Numbers gradually declined again to about 4,000 pairs in

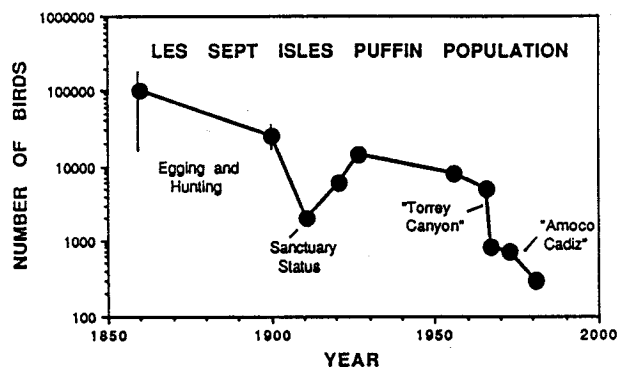


Figure 2. Population trends for Atlantic Puffins on Les Sept Isles, France. Note logarithmic abundance scale. Data from Yeatman (1976), Bourne (1976), Henry and Monnat (1981), Nettleship and Birkhead (1985).

1956 and to 2,500 pairs in 1966. The "Torrey Canyon" oil spill in 1967 reduced this remnant population to 400-500 pairs, and oil from the *Amoco Cadiz* spill in 1978 caused further declines up to 1981 when there were only 135-170 pairs (Bourne 1976, Hope-Jones et al. 1978, Henry and Monnat 1981). In 1984 bird numbers were estimated to have increased to 270 pairs (Mark Tasker, pers. comm.). This decline was paralleled by population reductions of Common Murres and Razorbills (*Alca torda*), both of which were also killed in large numbers by the 1967 and 1978 oil spills and now are almost extinct at Les Sept Isles (Nettleship and Evans 1985).

This example provides the best direct evidence for the effect of oil on seabird populations and typifies the situation for heavily polluted regions like the English Channel. At one time, significant populations of puffins, murres and razorbills bred on islands in the English Channel and along the bordering coasts of England and France. Many colonies have been extirpated, and remaining numbers are small relics of former populations (Nettleship and Evans 1985).

Newfoundland: Cape St. Mary's

Cape St. Mary's is the southernmost major breeding site for Common Murres in the western North Atlantic and the most southerly breeding site in the world for Thick-billed Murres (*Uria lomvia*; Tuck 1961, Nettleship and Evans 1985). Numbers of murres were estimated to be about 5,000 pairs in 1934 (Wynne-Edwards 1935), 2,500 pairs in 1942 (Peters and Burleigh 1951), and about 2,800 in 1959 including 285 pairs of Thick-billed Murres (Tuck 1961). By 1980 the population had increased to about 10,000 pairs of Common

Murres and 1,000 pairs of Thick-billed Murres (Nettleship 1980) paralleling a rapid growth of murre and other species populations at several other Newfoundland-Labrador seabird colonies (Nettleship 1980, Nettleship and Evans 1985, Montevecchi and Tuck 1987). Permanent census plots were established in 1980 to monitor the numbers of murres attending breeding ledges at Cape St. Mary's, and the numbers of immature murres attending "clubs" on the water have also been monitored annually since that time (Piatt and McLagan 1987, J.F. Piatt, D. Morrow, D.N. Nettleship, unpubl. data).

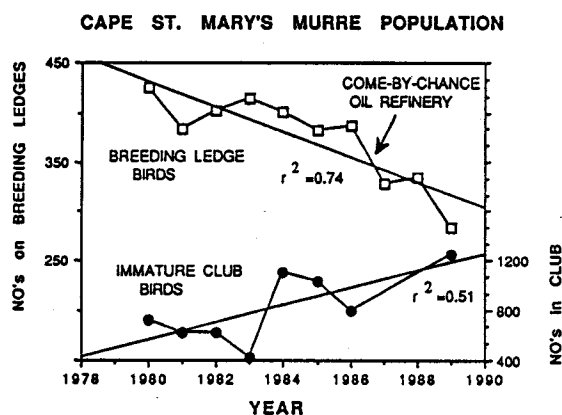


Figure 3. Population trends for Common Murres on breeding ledges and in an immature "club" at Cape St. Mary's Seabird Sanctuary, Newfoundland. Data from Piatt and McLagan (1987), J.F. Piatt, D. Morrow, D.N. Nettleship, unpubl. data.

The count data for murres show that the number of birds attending breeding ledges declined by 33% between 1980 and 1989, whereas the number of immature birds attending clubs has increased (Fig. 3). High breeding success of murres in 1980-1984 (5-year mean: ca. 0.70 fledglings/pair; J.F. Piatt, unpubl. data) and the presence of a large immature population suggests that the decline of breeding birds was due to unusually high adult mortality. The opposing trends in numbers of adult and immature murres at the colony suggest that increased adult mortality is being compensated for by increased survival and recruitment of immatures. It appears, however, that this compensatory effect is insufficient to offset the rate of loss of adults. The bycatch of murres in cod gill-nets was probably a serious source of adult mortality for birds at Cape St. Mary's throughout the 1980's (Piatt et al. 1984, Piatt and Nettleship 1987). However, oil pollution has also been a persistent source of mortality, adversely affecting birds in both summer and winter (Linegar 1979, Piatt et al. 1985, MacCharles 1986, J. Chardine, this volume). Since 1987, when

the nearby Come-by-Chance oil refinery was opened, there have been significant oil pollution incidents each winter and many thousands of murres are estimated to have been killed from single spills in March 1988, January-February 1989, and January 1990 (Canadian Wildlife Service-Atlantic files; P. Ryan, pers. comm.). Chronic low-level oil mortality has persisted over this period as well.

This example is representative of current alcid population trends in Newfoundland. As was the case in Britain, many seabird populations in Newfoundland expanded in numbers and range through the 1960's and 1970's, perhaps because of increased protection and optimal environmental and feeding conditions (Nettleship and Evans 1985, Montevecchi and Tuck 1987). With increased human competition for forage fish species (Nettleship 1977, Brown and Nettleship 1984a,b), and increasing losses from hunting (Elliot 1991), drowning in gill-nets, and oil mortality, populations appear to have either stabilized or started to decline. For example, attendance of Atlantic puffins at census plots on Great Island, Newfoundland, the largest colony in North America, has declined by 76% between 1974 and 1989—a change that represents at least a 30% reduction in the size of the breeding population (Nettleship and Evans 1985, D.N. Nettleship unpubl.). Similar declines are evident for a number of seaducks including the Common Eider (*Somateria mollissima*) and particularly Harlequin Ducks (Goudie 1989), the latter classified in 1990 as an endangered species in eastern Canada (R.I. Goudie, pers. comm.). Hunting and oil pollution are the most important mortality factors affecting wintering seaduck populations in Newfoundland and Atlantic Canada (Brown et al. 1973, Piatt et al. 1985, Reed 1986). The effects of oil pollution in Newfoundland extend to overwintering populations of Thick-billed Murres that may be declining at colonies in the eastern Canadian arctic (Nettleship 1977, Nettleship and Evans 1985), have declined dramatically in west Greenland (Evans 1991, Kampp 1991), and which are subjected to a substantial hunting kill toll in Newfoundland (Elliot 1991).

Central California: The South Farallon Islands

The history of Common Murre colonies on the South Farallon Islands off central California has been well documented (Ainley and Lewis 1974, Carter 1986, Ainley and Boekelheide 1990, Carter et al. 1990, Takekawa et al. 1990). Incidents of oil pollution affecting seabirds in this region are also well documented, particularly since 1970 (Dawson

1911, Aldrich 1938, Smail et al. 1972, Point Reyes Bird Observatory [PRBO] 1985, Stenzel et al. 1988, Page et al. 1990).

Up to 400,000 murres may have bred at the South Farallones before the Gold Rush in the mid-1800's (Fig. 4). Between 1854 and 1880, 12 million murre eggs were harvested to feed a booming market in San Francisco, and eggng by lighthouse-keepers continued even after it was banned at the turn of the century. By 1910, fewer than 20,000 murres remained on the South Farallon Islands. During the next 40 years the murre population declined further because of chronic oil pollution outside San Francisco Bay. Oil tankers routinely flushed their tanks before entering the bay, and several massive spills (e.g., following the "Frank H. Buck" collision in 1937) led to numerous reports of thousands of dead, oiled alcid's strewn on mainland beaches (Ainley and Lewis 1974). Chronic pollution continued through and after the war years (1939-1945), and only 6,000 murres were present at the South Farallon Islands in 1959. The demise of sardine (*Sardinops caerulea*) stocks in the 1940's probably also contributed to the suppression of murre numbers. Populations escaped major oil pollution incidents through the 1960's. In the late 1960's, colonies came under increasing protection and study by the U.S. Fish and Wildlife Service and Point Reyes Bird Observatory.

The response of the Common Murre population to more favorable conditions was dramatic (Fig. 4). Despite significant mortality from an oil spill in 1971 (Smail et al. 1972), there were probably about 45,000 murres breeding on South Farallon Island by 1972 (based on a boat count of 22,000 individuals and adjusted later from experience in comparing boat-based, aerial, and photographic census techniques). Growing at a rate of about 10% per annum, the South Farallon murre population peaked at 102,110 birds in 1982. Chances for a full recovery to historic levels were dashed, however, by a series of natural and anthropogenic influences between 1979 and 1990 (Carter 1986, Takekawa et al. 1990). A prominent El Nino-Southern Oscillation (ENSO) event in 1982-1983 reduced the productivity of murres and other seabirds in central California. Egg-laying by murres was delayed and reduced in 1983, and overall productivity was near zero (Ainley and Boekelheide 1990), although adult mortality remained the same as in previous years. Gill-nets set in Monterey Bay, Bodega Bay, and the Gulf of the Farallones killed a minimum of about 70,000-75,000 murres between 1979 and 1987 (by which time the problem had been largely eliminated with strict fishery regulations). This level of re-

corded mortality accounts for about half of the observed decline in numbers at the Farallon Islands and other central California colonies between 1980 and 1986 (Takekawa et al. 1990). Two major and several smaller oil spills caused mortality of Common Murres in central California after 1979. In November 1984, the "Puerto Rican" spill killed an estimated 1,500-2,000 murres in the Gulf of the Farallones area (PRBO 1985). In February 1986, the "Apex Houston" spill killed a minimum of 7,500 murres (Page et al. 1990). These spills seemed to have affected mainly central California murre populations. Significant chronic oiling also occurred through the period 1979-1990, but there are no statistically meaningful estimates of the actual kill of seabirds (Stenzel et al. 1988, PRBO unpubl. data).

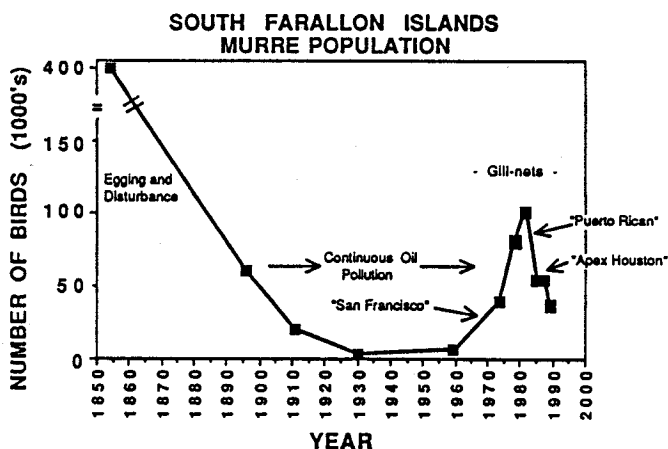


Figure 4. Population trends for Common Murres on South Farallon Island. Data from Ainley and Lewis (1974), Carter (1986), Ainley and Boekelheide (1990), Takekawa et al. (1990), Carter et al. (1990).

The marked declines of murres recorded at the South Farallon Islands and other central California colonies since 1982 did not occur at colonies in northern California (Fig. 5). Because the ENSO event affected all murre colonies in California in a similar manner, it seems unlikely to be an important factor in the decline of bird populations observed in central California. Most of the declines, and extinction of the small colony at Devil's Slide Rock, can be attributed to losses in gill-nets and from oil pollution (Takekawa et al. 1990). Reduced productivity and population sizes in the late 1980's will undoubtedly lead to further declines and prevent population recovery in the

near future.

From 1986 to 1989 the central California population declined by 7% even though mortality from drowning in gill-nets was reduced significantly by 1987 (see above). However, chronic oil pollution continues to kill thousands of murres each year (Carter, unpubl.data). Altogether, this case history illustrates how quickly seabird populations respond to changes in their environment, and how natural and anthropogenic mortality factors can act in concert to cause population declines and extinctions.

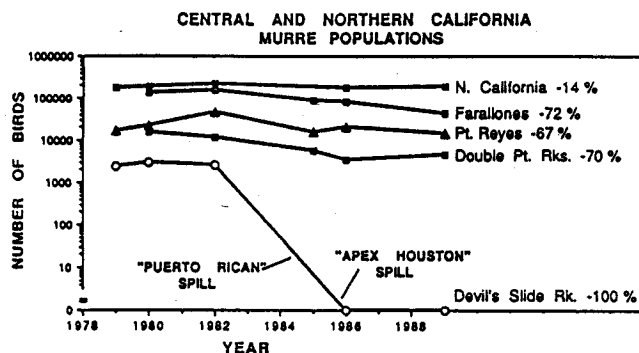


Figure 5. Population trends for Common Murres in central and northern California. Note logarithmic abundance scale. Data from Takekawa et al. (1990), and Carter et al. (1990). Northern colonies include Castle Rock, False Klamath Rock, and Flatiron Rock. Farallones estimates from north and south islands combined.

DISCUSSION

General conclusions

Although no one disputes the widespread and persistent mortality of marine birds from oil pollution, there is still some debate as to whether this mortality is of biological importance to populations. Opinions comprise the full range of possibilities, which vary from:

"To summarize, the species most exposed to, and most affected by oil pollution are auks and various seaducks, and oil pollution has probably played a major part in the decline in numbers of some species in some areas. ... In conclusion we can be fairly confident that... the long-term effects of ... oil pollution ... may be very serious for certain populations of seabirds." (Croxall 1975).

to:

"There is no evidence that seabird populations are declining as a result of oil spills. In fact, North Atlantic populations of most species have been increasing in recent years despite heavy annual losses from oil pollution. ... There is no reason to suppose that, from a biological point of view, this mortality is damaging to seabird populations". (Baker et al. 1990)

All investigators agree that where populations are stressed or declining, oil mortality can be a biologically significant mortality factor (Croxall 1975, Bourne 1976, Dunnet 1982, Clark 1984, NRC 1985, Camphuysen 1989). The degree to which oil mortality is additive, and thus is biologically significant over the long term for populations at carrying capacity or for increasing populations, has yet to be determined. However, the issue of carrying capacity rarely enters into the debate (but see NRC 1985). Many of the colonies for which there are reliable population data are today only a small fraction of their former size (Nettleship and Evans 1985) and population increases should be viewed from that perspective. In any event, the "no-effects" argument, which was based largely on the British North Sea experience of the 1970's, is now less attractive because many British seabird populations in that region are again declining (Stowe 1982b, Harris and Wanless 1988, 1990, Harris 1991) and mortality from oil remains undiminished (Mark Tasker, pers. comm.).

Seabird population declines or extinctions may be defined clearly as significant biological effects, especially for species that exhibit a strong fidelity to traditional breeding sites. Such breeding areas may be irreplaceable and, unless recolonization occurs, may result in a permanent reduction in productivity of a regional population. There are, however, other considerations in assessing the importance of oil mortality. For instance, the extinction of a single colony or sub-population may not by itself be significant on a regional or global basis, but the cumulative effect of a series of such events might be catastrophic to a species population over the long term. Moreover, small populations, or "sub-populations" may play an important role in conserving biological diversity and maintaining reservoirs of genetic information. They may also act as foci for re-establishing populations following major perturbations or where ameliorating environmental conditions allow population expansion (Cairns and Elliot 1987). More subtle effects may include permanent changes

in local community structure. Many of the species affected by oil are dominant members of northern marine food webs, and large one-time losses may have positive implications for competitors and result in long-term alterations of local food-web and breeding colony dynamics (NRC 1985). Finally, there are socio-economic considerations. For example, the extinction of puffins at Sept Isles, France, murres at Cape St. Mary's, Newfoundland, or murres at the Farallon Islands, California do not endanger the species over their world range, but the citizens of France, Canada, and the United States place immeasurable aesthetic value on those seabirds. All of these localities are also tourist attractions and devastation of these colonies would translate directly into lost aesthetic, educational, and economic values.

It is not coincidental that most of the evidence presented and discussed here for population effects of oil pollution are derived from studies on alcids, particularly Common Murres at the southern extremes of their range in the Atlantic and Pacific oceans. The Common Murre is one of the dominant marine birds in northern regions, and as a colonially-breeding species that can be observed with relative ease on land during the breeding season it functions as a valuable indicator species of marine environments. Evidence for effects on other avian taxa (loons, grebes, seaducks) or on alcids breeding in inaccessible high-latitude regions are lacking, largely because we cannot easily monitor those populations. Without further qualification, however, it is clear that over decades of study, many investigators have reported significant damage to local or regional populations of non-alcid species from oil pollution (e.g., Curry-Lindahl 1960, Greenwood 1970, Joensen 1972a, 1972b; Soikkeli and Virtanen 1972, Brown et al. 1973, Campbell et al. 1978, Heubeck and Richardson 1980, Piatt et al. 1990, and see reviews by Bourne 1976, Hooper et al. 1987, Camphuysen 1989). Although the present review is restricted to marine birds in the northern hemisphere, it should be noted that oil pollution also has affected many species in the southern hemisphere, particularly penguins and other divers (Westphal 1969, Ross 1971, Bourne 1976, Hooper et al. 1987).

Compared to Europe, far less data are available for assessing the effects of oil pollution on present-day marine bird populations in North America. However, given the prevalence of bird mortality from chronic oil pollution around Newfoundland (Piatt et al. 1985, MacCharles 1986, J. Chardine, this volume) and along the eastern Canadian and U.S. seaboard (Brown et al. 1973 and this volume, Simons 1985); declining or threatened populations of alcids, seaducks and

loons in many regions (Nettleship and Evans 1985, Simons 1985, McIntyre 1986, Reed 1986, Goudie 1989, Elliot 1991, Evans 1991); current development of the Hibernia oil field on the Grand Banks of Newfoundland and continuing oil exploration in offshore areas of eastern Canada; we expect that oil pollution, acting in combination with other mortality factors, will continue to negatively affect bird populations in the northwest Atlantic region for decades.

Evidence for the negative effect of oil pollution on murre populations in central California is compelling (Stenzel et al. 1988, Takekawa et al. 1990, Page et al. 1990). It is likely that Marbled Murrelet numbers are declining in California because of losses from oil pollution and the continued removal of old-growth forest breeding habitat. Continued decline of the Marbled Murrelet, possibly leading to extinction, has been projected (Carter and Erickson 1988). Oil pollution has also been implicated in limiting the population recovery of other alcids in central California (Ainley and Lewis 1974). Endangered California Brown Pelicans (*Pelicanus occidentalis californicus*) may be similarly affected by chronic oil pollution (Anderson and Fry, this volume). Little is known about population effects of oil pollution in other west coast areas, although oil pollution is chronic in British Columbia and many local populations are vulnerable (Vermeer and Vermeer 1975). The "Nestucca" oil spill off Washington in December 1988 provides some indication of the magnitude of damage from oil pollution in this region. More than 13,000 oiled birds comprising 31 species were recovered on beaches (Rodway et al. 1989), close to 10,000 in Washington (80% Common Murres) and about 3,000 along the west coast of Vancouver Island (42% Common Murres, 32% Cassin's Auklet; A.E. Burger, pers. comm.). If we assume a conservative recovery rate of 30% (Table 1), then about 40,000 birds may have died in this single incident.

Owing to the remoteness of bird populations in Alaska and the sparse human population, the kill toll from oil pollution in this region has been poorly documented. Two spills in 1970 may have each killed about 100,000 seabirds, mostly murres (McKnight and Knoder 1979). Following the *Exxon Valdez* spill in Prince William Sound, about 30,000 oiled birds (74% murres) were recovered and the actual kill toll was conservatively estimated at 100,000-300,000 birds (Piatt et al. 1990). About 10% of the Gulf of Alaska Common Murre population may have been eliminated, and local populations of Marbled Murrelets, Harlequin Ducks, loons, grebes and other species in the path of the spill (30,000 km² area) were decimated (see

also McIntyre, this volume). Limited beach survey data suggests that oil mortality is a chronic problem in Alaska (U.S. Fish and Wildlife Service, unpubl. data). In 1988 and 1989 alone, 43 oil spills involving 14 million gallons of oil were reported in Alaskan waters (including 11 million gallons spilled by the *Exxon Valdez*; U.S. Coast Guard, unpubl. data). Several of these spills were in the vicinity of major seabird colonies, but damages were not documented. Chronic pollution is likely to get worse in Alaska as fishing fleets expand and more oil exploration and development occurs in offshore environments (Lensink 1984).

The overall effect of chronic oil pollution in Alaska are unknown. It seems likely, however, to be important for certain breeding populations already stressed by other factors. For example, productivity of Common Murres, Black-legged Kittiwakes (*Rissa tridactyla*), and several other piscivorous seabirds in Alaska is generally much lower than in north Atlantic populations, and populations are declining at several colonies (Byrd et al. 1985, Springer et al. 1985, Hatch 1987, Piatt et al. 1988, Harris and Wanless 1988, Fadely et al. 1989, Hatch and Hatch 1990, Hatch et al. 1991, Springer 1991).

Future Research

Four critical data gaps must be filled if meaningful assessments are to be made on impacts of oil pollution on bird populations in North America and elsewhere:

(1) Monitoring breeding colonies- we must continue, improve, and expand our efforts to monitor populations known to be affected by oil pollution. This requires a major commitment of resources by government agencies responsible for the conservation and protection of marine resources. However, the information obtained on changes in numbers and status of seabirds is also essential for monitoring effects of other anthropogenic mortality factors and effects of regional and global changes in marine ecosystems.

(2) Beached-bird surveys- we need to expand and increase beached-bird surveys in all regions of North America to establish baseline trends for levels of chronic pollution and to assess effects of new petroleum developments offshore, occasional catastrophic spills, and changes in levels of chronic pollution. This effort requires only a modest commitment from managing agencies if surveys are coordinated using volunteers and conservation groups (e.g., Simons 1985, Stenzel et al. 1988).

(3) Oil-spill response- we need specific contingency plans to provide for carcass counts on beaches and at rehabilitation centers following oil spills so we can document accurately the numbers of birds reaching shore (Carter et al. 1987, Carter and Page 1989, Page et al. 1990).

(4) Oil-mortality assessment- we need better estimates of the proportion of birds killed by oil at sea that do not wash ashore following chronic and large-scale oil spills. This will require refinement of aerial survey techniques to measure populations at risk at the time of a spill and more research on model parameters used to integrate oil slick movements, bird distribution, and carcass trajectories. Carcass drift experiments, conducted during actual oil spills, would greatly improve our ability to estimate total bird losses from oil pollution.

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Table 1. Drift experiments to determine the proportion of birds killed at sea that are detected on beach surveys.

Region	N	Species or model	Recovery %	Reference
Bristol Channel	?	Gulls	10.0	Beer in: Bibby & Lloyd 1977
Irish Sea	300	Gulls	11.0	Bibby & Lloyd 1977
Irish Sea	305	Gulls	44.0	Bibby & Lloyd 1977
Irish Sea	347	Gulls	59.0	Bibby & Lloyd 1977
English Channel	144	Gulls	20.0	Hope-Jones et al. 1978
S. North Sea	600	Gulls	9.8	Bibby 1981
S. North Sea	40	Gulls	40.8	Stowe 1982c
S. North Sea	150	Gulls	11.3	Stowe 1982c
California	186	Gulls*	29.9	Page et al. 1982
Mean for <i>Larus</i> gulls			26.0	
California	63	Alcids*	0.0	Page et al. 1982
N. Grand Banks	115	Alcids	0.0	Threlfall & Piatt 1983
N. Grand Banks	400	Blocks**	0.0	Threlfall & Piatt 1983
S. Grand Banks	129	Alcids	0.0	Threlfall & Piatt 1983
S. Grand Banks	600	Blocks**	24.0	Threlfall & Piatt 1983
Gulf of Alaska	100	Alcids	3.0	Piatt et al. 1990
Irish Sea	410	Alcids	20.0	Hope-Jones et al. 1970
Irish Sea	319	Alcids, Gulls	7.5	Lloyd et al. 1974
Mean for Alcids			5.0	

* Gulls and alcids released together in the same experiments.

** Wood blocks released at the same times and locations as birds.

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